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Literature Review for the National Uranium Tailings Program (NUTP) Physical Parameters of Tailings and Underlying Aquifer Study on Contaminant Migration

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SUMMARY

A review of the published literature and personal communication with experts in the field of tailings physical property studies and modelling was conducted to attempt to clarify and/or better understand certain physical parameters of tailings and their effect on the removal of radionuclides and other contaminants associated with uranium mill tailings. The main points of interest of this review are:

- Groundwater flow rates
- Contaminant flow rates
- Hydraulic gradient
- Porosity
- Density
- Diffusivity
- Dispersion
- Soil temperature

These points, the effect of their variations with time over the long-term and some specific erosion processes are discussed.

S O M M A I R E

Une revue, de publications et de communications personnelles, au sujet de l'étude des propriétés physiques des résidus d'uranium et de la modélisation a été effectuée. Ceci dans le but d'éclaircir et d'approfondir nos connaissances de certains paramètres physiques des résidus d'uranium et de l'influence de ces paramètres sur la migration de radionucléides et autres contaminants associés aux résidus d'uranium. Les principaux points d'intérêt de cette revue sont:

- la vitesse d'écoulement des eaux souterraines
- la vitesse de migration des contaminants
- la conductivité hydraulique
- le gradient hydraulique
- la porosité
- la densité
- la diffusivité
- la dispersion
- la température des sols

Ces points, ainsi que d'autres tel que les considérations de l'effet du temps à long-terme et certains types d'érosion, sont discutés.

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1. INTRODUCTION

All ore mining and milling operations produce wastes, called tailings, that can vary in their environmental impact. Mining and milling of uranium ore is no exception. By 1982, there were about 131 million tonnes of uranium mill tailings in Canada.

Although uranium mill tailings make up only a small proportion of the total tailings produced in Canada, interest in their environmental effects is widespread because of their radioactive content. However, there is little information about the interaction of these wastes with the environment, particularly over long periods.

Some questions have been raised about the long-term effects on the environment after mining and milling at the sites has ended.

That is why in 1981, after a year-long study, the National Technical Planning Group on Uranium Tailings Research concluded that there was not enough scientific data to predict confidently the long-term effects (1). This group of experts, appointed by Energy, Mines and Resources Canada, recommended that a research program be conducted to fill the knowledge gap.

Thus, in 1982 the Government of Canada announced the National Uranium Tailings Program, a five-year research program to focus on predicting the effects on the environment of uranium mine tailings and the effluent from them over the long-term. To assist the Program, a Technical Advisory Committee that includes representatives from industry, and the federal and provincial governments was formed and has held five meetings to date. It was in one of these meetings that the question of the importance of physical parameters of tailings was raised. This was the reason that the measurement program asked for a literature review on the physical parameters of tailings and underlying aquifers and a study on contaminant migration.

2. GROUNDWATER FLOW AND CONTAMINANT RETARDATION

2.1 GROUNDWATER FLOW

Throughout the review of the literature, it has been shown that the physical parameters of the uranium tailings and the physical parameters of the surrounding sand aquifers are not the primary factors influencing the migration of contaminants such as Ra-226, U, Th-230, Pb-210, many heavy metals and the low pH fronts. The physical parameters are, however, the primary factors that influence porewater flow rates and groundwater flow rates. The major causes of retardation are geochemical processes.

For instance, in deposits of sand or gravel, if contaminants are not removed from solution by geochemical processes, they are transported by the flowing groundwater at approximately the average rate of groundwater flow which, for sand or gravel, is in the range of about 0.1 m to 10 m per day (2). Transport without retardation is referred to as advection. Simple advection of contaminants is not known to exist in and around uranium tailings impoundment areas because of the many geochemical reactions that occur in the porewater and groundwater regimes.

2.1.1 Nordic Tailings Site

At the Nordic tailings site the average lateral specific groundwater discharge in the surficial aquifer is of the order of 20 $m^3/m^2/yr$ and the average seepage velocity is 70 m/yr. These values increase towards the toe of the tailings dam where the hydraulic gradient is larger. The rate of the southward flow of groundwater in the plume near the toe of the dam is approximately 550 m/yr and the specific discharge from this area is an average value of 167 $m^3/m^2/yr$. The specific discharge and velocities rapidly diminish southward from this point (2, 3). In the Nordic tailings the groundwater flow is downward at a velocity of about 0.5 to 1.5 m/yr. A general description of the Nordic tailings site is given in Appendix "A".

Calculated groundwater velocities in the upper sand layers using the hydraulic conductivity - hydraulic gradient relationships yield values of about 440 m/yr near the dam (at M-1), about 160 m/yr at M-9 and about 40 m/yr at M-10 and M-14. A cross-section of the Nordic main tailings dam giving the locations of the piezometers is shown on Figure 1.

2.1.2 Williams Lake Tailings Site

At the Williams Lake tailings site, the small depth of penetration corresponds to vertical porewater velocities of 0.07 m/yr to 0.23 m/yr. Such velocities are comparable to the porewater velocities estimated from hydraulic conductivity - hydraulic gradient relationships (2). A general description of the Williams Lake tailings site is given in Appendix "A".

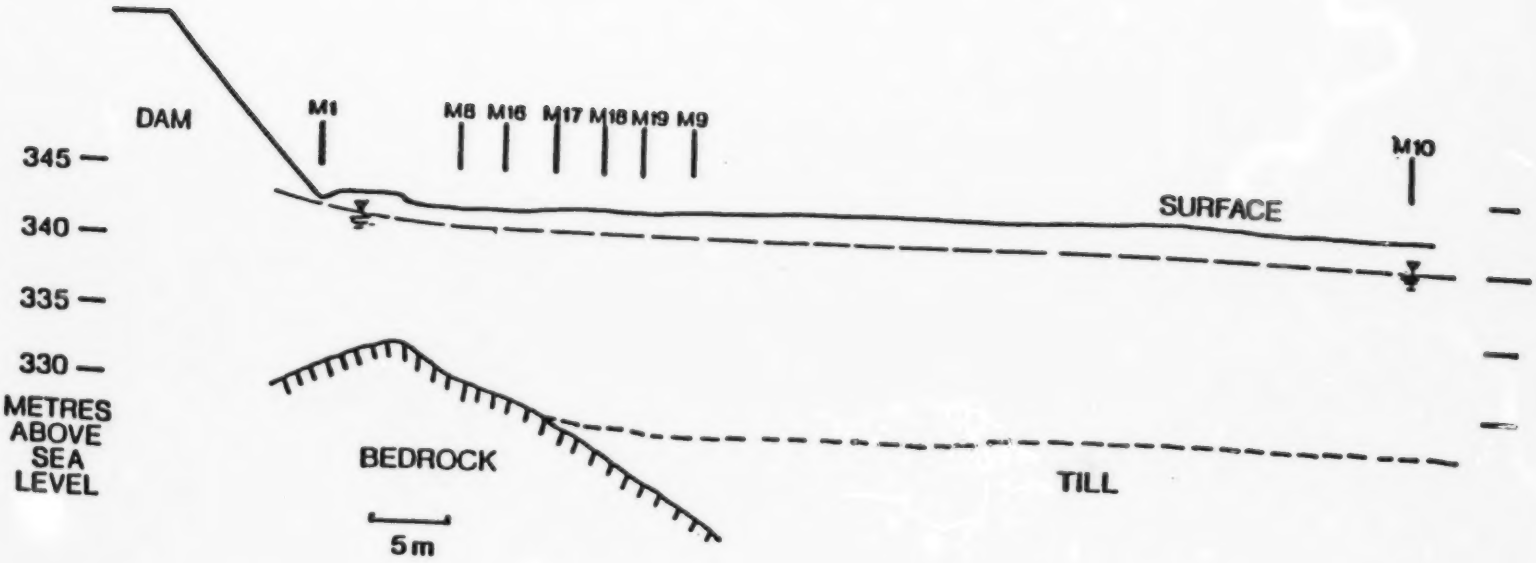


Figure 1. Piezometer locations at the Nordic main tailings site.

2.1.3 Measurement

Because of the uncertainty in the hydraulic conductivity measurement (see Section 3.1), the calculated velocity may be different from the actual velocity by a factor of 2 or more. Therefore, the velocity near the dam at the Nordic tailings site is probably between 250-1000 m/yr, with velocities in the lower part of the range more probable (4).

2.2 RETARDATION

The retardation is due to geochemical processes that causes considerable neutralization of acidic water in the groundwater zone and causes the zone fronts of the acidic groundwater to move at rates much slower than the groundwater flow rates. The advance of the low pH zone occurs at a rate that is only 0.1 to 0.3 per cent of the groundwater flow rate (2). Contaminant such as Ra-226 and heavy metals have also migrated in the aquifer at much slower velocities than the groundwater velocity.

The retardation of the movement of the acid front is attributed primarily to the dissolution of calcite that exists in the sand aquifer. The sand contains less than one weight per cent calcite. As calcite dissolves in the neutralization zone siderite ($FeCO_3$) and gypsum ($CaSO_4 \cdot 2H_2O$) precipitate.

When tailings impoundments are located on sand deposits that have appreciable acidic-neutralization capacity, the degree of attenuation of tailings-derived contaminants is generally very high.

The field data indicate that pH is a dominant control on the behaviour of metals and radionuclides in the sand aquifer. This control is apparently exerted by way of the strong pH dependency of precipitation and adsorption reactions.

Heavy metals and Ra-226 will be relatively immobile in the permeable sandy deposits below and beyond the tailings except if the mobility is enhanced in zones where the pH of the groundwater declines below about 5.

A very small percentage of calcite in the tailings can provide a large neutralization capacity for acid produced by pyrite oxidation. For example, if porewater with an initial pH of 3 passes through a unit volume of saturated tailings with a porosity of 0.33 and a calcite content of 0.15 per cent and if the acid reacts rapidly with the calcite, approximately 80 pore volumes of the acidic water would need to pass through the porous medium before all of the calcite was removed. In other words, under these conditions the front of acidic water would advance through a column of the porous medium at a velocity equal to 1/80 of the average linear groundwater velocity (3).

2.2.1 Groundwater Flow and Retardation at the Nordic Tailings Site

To obtain estimates of the degree of retardation of Ra-226 and heavy metals, which are limited in occurrence to an area very close to the toe of the Nordic tailings dam, the groundwater velocity must be estimated specifically for this area. For example, a representative value for the hydraulic conductivity of the coarse, clean sand that comprises the aquifer at the Nordic site is 1×10^{-2} m/s (4).

This indicates that the Ra-226 plume advances at a rate of about 0.5 per cent of the average linear groundwater velocity. The frontal positions of the zones of high concentration of Ni^{2+} and Co^{2+} correspond closely to the front of the Ra-226 zone. These contaminants also probably began their advance when seepage from the tailings near the toe of the dam became acidic. Thus, the retardation factor for the nickel and cobalt plumes is probably similar to the factor for Ra-226.

2.2.1.1 Measurement

There is considerable uncertainty associated with the retardation factors presented in the above section. The uncertainty arises because of uncertainty in the estimated value for the average linear groundwater velocity and, because of lack of precision of the location of the starting position of the zone of acidic water upon entry into the aquifer in approximately 1968. In spite of these uncertainties, however, there is little doubt that the rate of advance of the zone front of groundwater with low pH, high Ra-226 and heavy metals is only a small fraction of the average groundwater velocity.

2.2.1.2 Retardation by Calcite at the Nordic Tailings Site

This retardation will cease in later years with the loss neutralization capability probably because of removal of all of the calcite from the shallow part of the tailings or because of a reduction in effectiveness of the calcite that remains in the shallow zone. A loss of effectiveness may develop because of the formation of precipitate coatings of iron hydroxide on the calcite grains (3).

It is reasonable to expect that, as the acidic water from the tailings moves in the sandy aquifer, the acid is consumed by reaction with calcite in the aquifer. Analyses of the sand from a location in the aquifer in front of the low pH plume gave values ranging from 0.008 to 0.14 per cent calcite. Thus, for porewater with an initial pH of 4, between about 5 and 860 pore volumes, respectively, would have to pass through the medium to remove all the calcite. If the calcite is removed along a sharp front, which forms the boundary between a zone where all calcite has been removed and a zone where none has been removed, the front would advance at a rate of 1/5 of the groundwater velocity if the calcite content is 0.008 and at 1/860 if the calcite content is 0.14 per cent (3).

The above examples serve to illustrate that a very small percentage of calcite in the sand could cause a strong degree of retardation in the zones of low pH water, radium-226 and other heavy metals.

More calcite content analyses for the sandy aquifers will be necessary before representative values can be used in calculations of this type (3).

2.2.1.3 Retardation by Sorption in the Nordic Plume

Another likely cause of retardation of the zone front of Ra-226 in the aquifer is sorption. Ra-226 is dissolved to a much greater degree at higher pH. The position of the zone front of high Ra-226 would, therefore, be controlled directly by the position of the transition zone between lower and higher pH waters. The relative degrees of influence of sorption and co-precipitation on the retardation of Ra-226 cannot be determined with the existing data (3).

2.2.1.4 Retardation by Calcite in the Nordic Plume

A very small but significant percentage of residual calcite in the tailings has prevented much of the zone of recharged water from becoming strongly acidic as a result of pyrite oxidation.

Blair et al (3) attributed the retardation of the inner core advance of the plume to chemical buffering by solid phase carbonate such as calcite ($CaCo_3$) in the aquifer. Three sand samples analysed by Blair for carbonate mineral content gave values from 0.008 to 0.14 per cent. Thus, it is reasonable to expect that the front of the low pH inner core is neutralized as it advances and, therefore, the movement of the low pH front is retarded with respect to the groundwater velocity.

Total consumption of calcite is 2.61 moles per cubic metre of aquifer (4). When this consumption is considered in the neutralization zone, an inner core velocity of about 9 per cent of the groundwater velocity is obtained. Although this result is based on the carbonate content of the lower sand, the carbonate content of the upper sand is probably close to the retardation factor obtained by comparison of the 1979 and 1980 data (4). This data indicates that the inner core is moving at a very small percentage of the groundwater velocity. This retardation is the result of pH neutralization by dissolution of calcium carbonate.

2.2.2 Conclusions on Retardation

The preceeding pages indicate that retardation, mainly caused by calcite reactions in the tailings and in the aquifer, is well documented in the literature. Retardation factors of 0.1 to 0.3 per cent are also generally accepted. These factors are important considerations for modelling. The effects of physical parameters on groundwater and porewater flow rates and, thus, on retarded contaminant flow rates are important.

They are used to calculate the time a pore volume is capable of neutralizing the low pH front. Although there is a retard flow, it is still a function of groundwater or porewater velocity. If the flow rate increases, the retarded flow rate will also increase. Consequently, the time frame can be effected by the physical parameters that affect flow rates.

3. PHYSICAL PARAMETERS

3.1 HYDRAULIC CONDUCTIVITY

Hydraulic conductivity is defined as being the flow rate of water through the tailings or the saturated percolation rate (2). For the purposes of this program, hydraulic conductivity is theoretically well understood and is easily measured.

Hydraulic conductivity "K" is sometimes called the coefficient of permeability. The permeability "k" is the facility with which the water can move through the earth mass. The "K" value is proportioned to the "k" value (5). There are many different equations that exist for calculating the hydraulic conductivity (5, 6). It can be calculated using many different parameters such as density, specific gravity, grain size distribution, moisture content, porosity and pressure head. It can also be affected by low temperatures creating frost and permafrost conditions that changes the value of the viscosity property of the tailings (5).

Some authorities feel that data on hydraulic conductivity and moisture content for uranium tailings would be required to carry out their investigations (7).

Before starting on such work, it would be better to determine how widespread this demand is and the specific reasons for obtaining this data.

3.1.1 Hydraulic Conductivity at the Nordic Tailings Site

Hydraulic conductivity values in the Nordic tailings reflect stratification by layers of different grain size. In general, the lower hydraulic conductivities are located near the bottom of the tailings and are overlain by tailings of higher hydraulic conductivities. This trend is best represented at piezometer nest T-2 where "K" values of 2×10^{-4} to 4.6×10^{-4} cm/s overlie "K" values of 3.1×10^{-6} to 6.7×10^{-6} cm/s. Locations of piezometer nests at the Nordic main tailings site are shown on Figures 2 to 4. This is thought to represent deltaic grading of coarser sand tailings out over finer slime tailings. The hydraulic conductivity of the peat layer beneath the tailings at T-1 and T-2 was calculated to be 4×10^{-7} cm/s (3).

Empirical methods using grain size distributions to calculate hydraulic conductivities were used at the Nordic main tailings site. The method (Masch and Denny) uses a range of grain sizes rather than one grain size (5). It is, therefore, more sensitive to characteristics of the entire grain-size distribution.

A total of six grain-size distributions were measured in samples obtained from the plume area: four samples of the upper sand and two samples of the lower sand. The arithmetic-mean of the hydraulic conductivity of the upper and lower sands were about 1.2×10^{-4} and 5.3×10^{-5} m/s respectively (4).

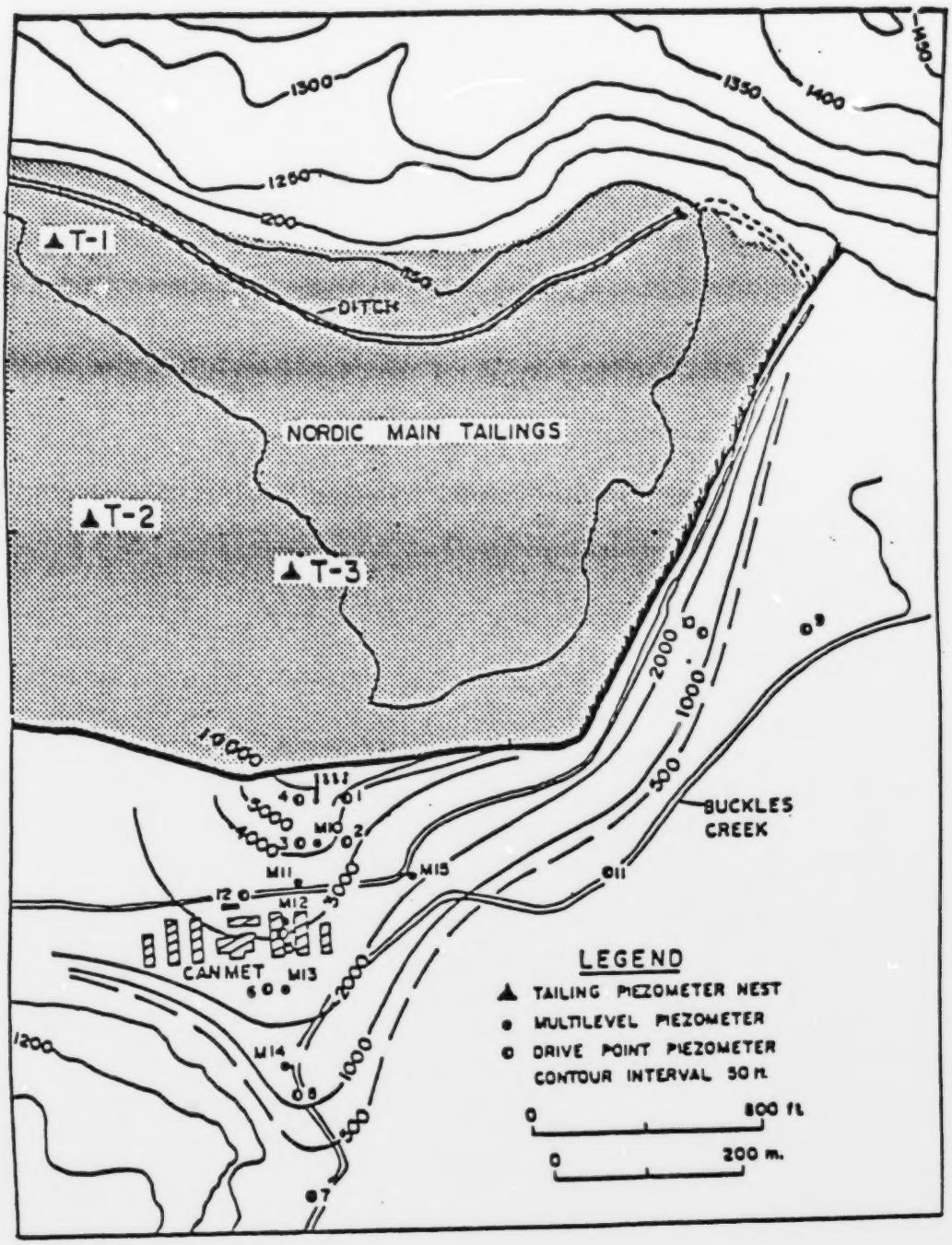


Figure 2. Topography of a study area at the Nordic main tailings site (3).

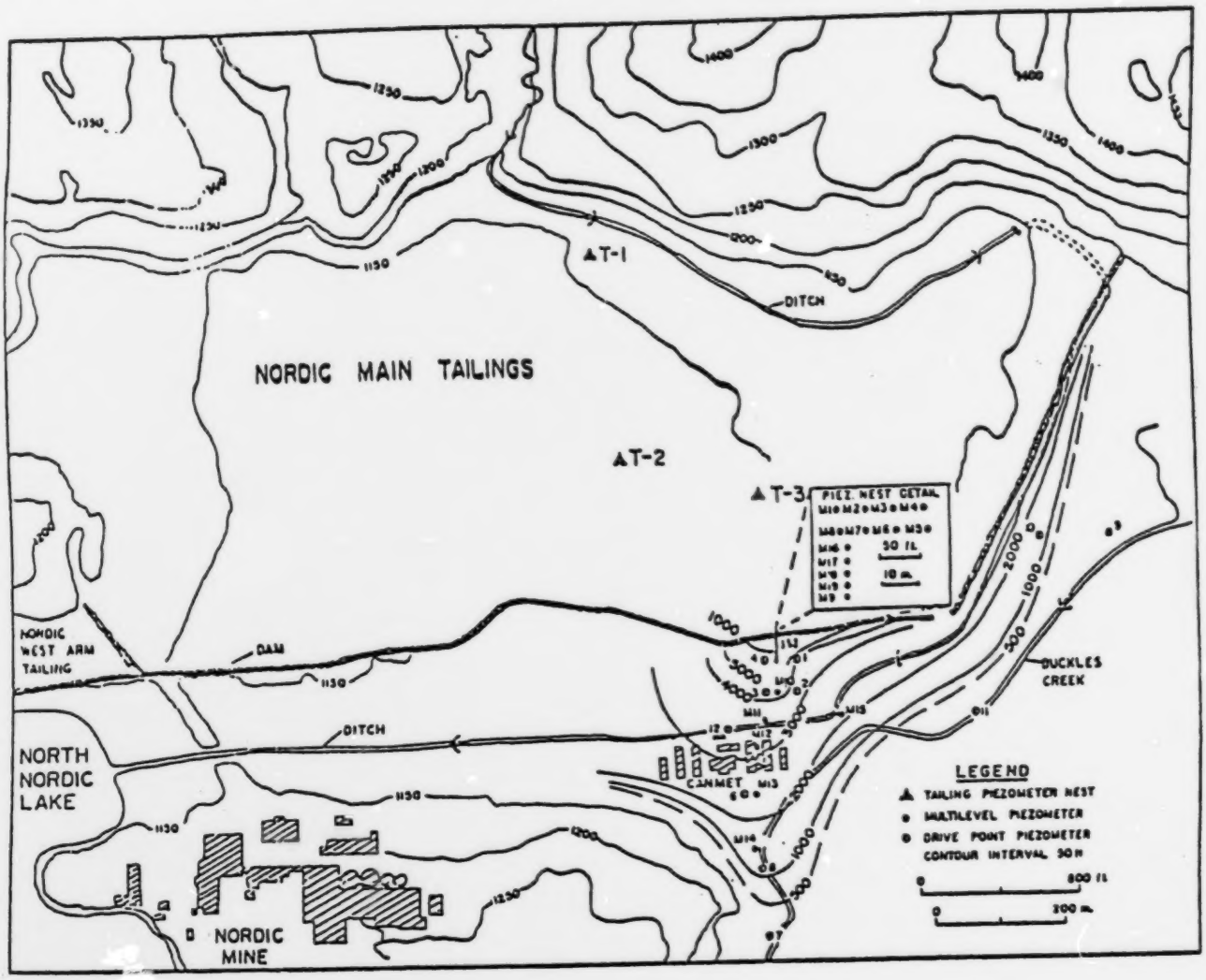


Figure 3. Groundwater instrumentation locations at the Nordic main tailings site (4).

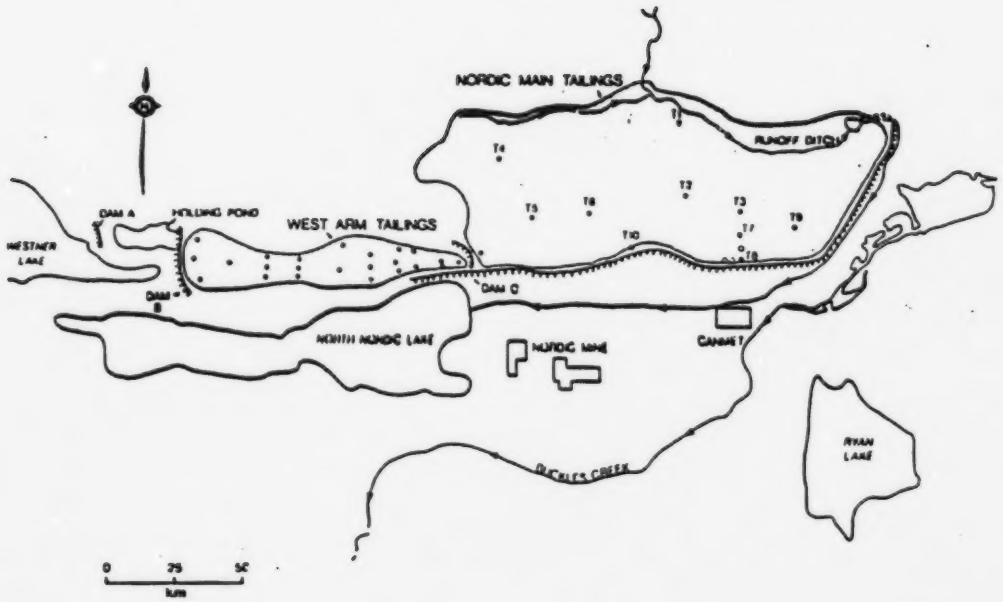


Figure 4. Location map and monitoring sites at the Nordic main and Nordic west arm tailing (9).

At the Nordic west arm tailings, the average value of hydraulic conductivity obtained was approximately $1x10^{-5}$ cm/s, which is in agreement with the average rising-head test values, and is considered representative of the average tailings hydraulic conductivity in an area of predominantly finer grained tailings distal to the tailings outfall.

3.1.2 Hydraulic Conductivity at the Williams Lake Tailings Sites

At the Williams Lake tailings sites, the hydraulic conductivity varies from 10^{-4} cm/s in sandy tailings to 10^{-6} cm/s in silty tailings as determined from piezometer response tests (2).

3.1.3 Discussion

Morin (4, 8) has said that there is uncertainty in the hydraulic conductivity values. The calculated velocity may be different from the actual velocity by a factor of 2 or more. Uncertainties of this order are seen as very low (i.e. good measurement) and uncertainties of up to 5 times are still considered acceptable.

Elaborate measurements are not generally required for conductivity determinations on field samples. The variability among samples is usually large enough that the precise determination of the hydraulic conductivity of a given sample is not warranted. It is sufficient to note that temperature differences for field and laboratory measurements can influence hydraulic conductivity values through the viscosity term. The effect is usually small, and correction factors are seldom introduced.

Uncertainties on hydraulic conductivity measurements, however, create similar uncertainties on flow rates (i.e. using Darcy's Law). It would be important to estimate groundwater velocities more accurately resulting in a more precise estimate of the retarded flow rates of contaminants.

3.2 HYDRAULIC GRADIENT

Hydraulic gradient is well defined in the literature. It is theoretically well understood and is easily measured.

Basically, the hydraulic gradient is the slope of the water table and, in turn, the water table is best defined as the surface on which the fluid pressure in the pores of a porous medium is equal to atmospheric pressure (5). The location of this surface is revealed by the level at which water stands in a shallow well open along its length and penetrating the surficial deposits just deep enough to encounter standing water in the bottom. If the fluid pressure (p) is measured in gage pressure, then at the water table p=o. The water table is commonly thought of as being the boundary between the unsaturated zone at the surface and the saturated zone at depth, yet we are aware that a saturated capillary fringe often exists above the water table.

In most uranium tailings, the hydraulic gradients are predominantly downward. However, the presence of a peat layer or a stratigraphy of coarse and fine grains can cause large changes in the hydraulic gradients.

3.2.1 Hydraulic Gradient at the Nordic Tailings Site

At the Nordic tailings site, the downward gradient is affected by the presence of a peat layer which causes a sharp increase in the gradient. The downward gradient in the tailings at piezometer nests T-1 and T-2 is fairly constant at 0.06 m/m then sharply increases to 1.45 m/m across the peat layer in response to a much lower hydraulic conductivity in the peat than the overlying tailings. Locations of piezometer nests at the Nordic tailings site are shown on Figures 2 to 4. The peat layer is not present at piezometer nest T-3 and the gradient is constant at 0.2 m/m through the tailings into the underlying gravel. The horizontal hydraulic gradient in the tailings and underlying gravel aquifer is 0.0065 m/m towards the south, approximately 3 orders of magnitude less than the vertical hydraulic gradient. The horizontal gradient component increases slightly towards the toe of the tailings dam. Representative horizontal gradients between the tailings dam at piezometer M-1 are listed on Table 1.

Piezometers	Hydraulic Conductivity* cm/s	Downward Hydraulic Gradient*	Downward Specific Discharge $m^3/m^2/year$	Downward Groundwater Velocity** m/year
T-1	$1x10^{-5}$	0.060	0.14	0.47
T-2	$1x10^{-5}$	0.058	0.14	0.46
T-3	$1x10^{-5}$	0.21	0.50	1.66

*Representative values
**Specific discharge (i.e. the Darcy velocity) divided by porosity

TABLE 1. Summary of groundwater flow parameters in the Nordic main tailings (3).

3.2.2 Hydraulic Gradient at the Williams Lake Tailings Site

At the Williams Lake tailings site, the water table follows the surface of the tailings sloping from east to west, and represents a horizontal hydraulic gradient of 0.5 per cent. The tailings porewater leaves the tailings site at the west end through the valve controlled drain in Dam No. 1. The vertical components of the hydraulic gradient in the tailings are predominantly downward and range from 1.5 to 2.5 per cent. The vertical hydraulic gradients at piezometer nest WL-3 beneath the headpond are upward when the headpond level is low but downward when the headpond level is high. The location of the piezometer nests at the Williams Lake tailings site are shown on Figure 5. The southward gradient of the water table is about 0.050 per cent near the dam, and decreases to about 0.005 per cent (the detection limit) approximately 20 m from the dam(2).

3.2.3 Hydraulic Gradient at the Stanrock Tailings Site

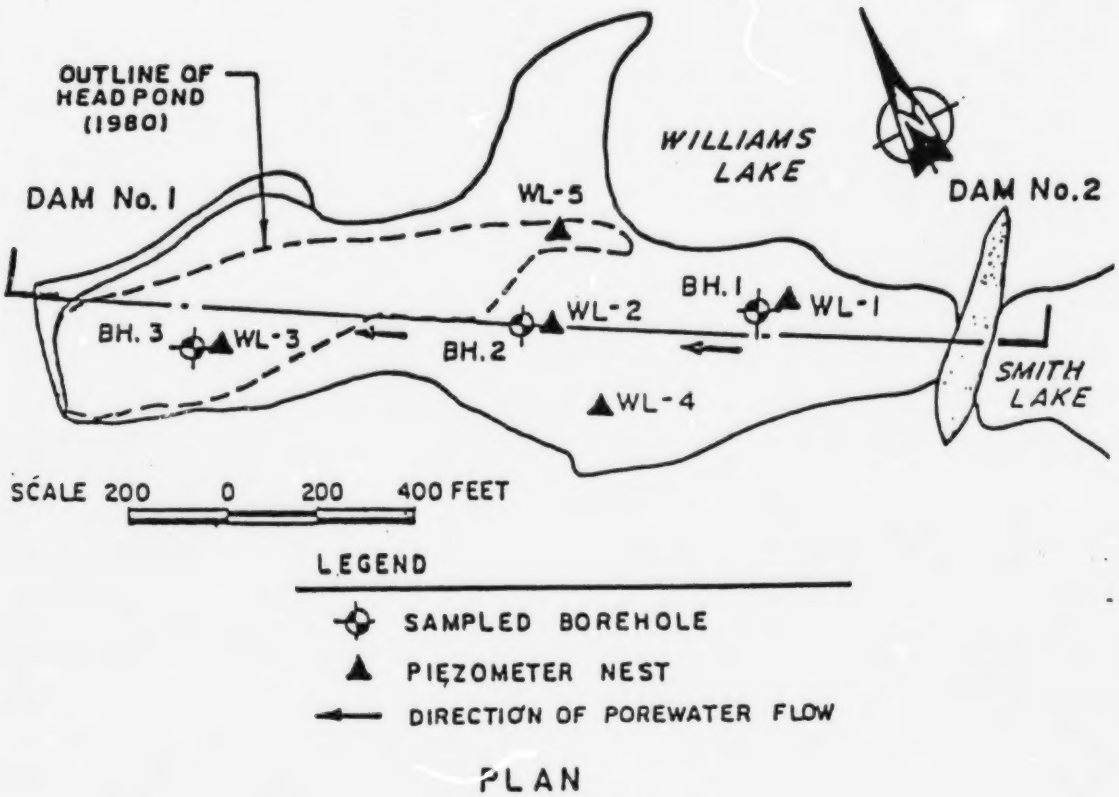
At the Stanrock tailings site the horizontal component of the hydraulic gradient is directed outward from the centre of the tailings area toward the pervious dams. A general description of the Stanrock tailings site is given in Appendix "A". The horizontal hydraulic gradient is of the order of 0.5 per cent near the centre of the basin but ranges as high as 15 per cent immediately adjacent to the pervious dams. The vertical components of the hydraulic gradient are everywhere downward, and range from 5 per cent near the centre of the tailings site to 50 per cent adjacent to Dam "A" (2). The Stanrock tailings site is shown on Figure 6.

3.2.4 Discussion

The extent of recharge water movement into the tailings is dependent on the nature of the tailings materials and the hydraulic gradient. Such velocities are comparable to the porewater velocities estimated from hydraulic conductivity - hydraulic gradient relationships.

At the Stanrock tailings site, the depths of penetration correspond to porewater velocities which are also comparable to the velocities estimated from hydraulic conductivity - hydraulic gradient relationships.

The pattern of recharge water penetration into the tailings identified at the Williams Lake and Stanrock sites at Elliot Lake by Cherry et al (9) are in areas where downward hydraulic gradients occur. At other inactive tailings impoundments, where the hydraulic gradients are upward, the recharge water does not penetrate into the tailings (9).



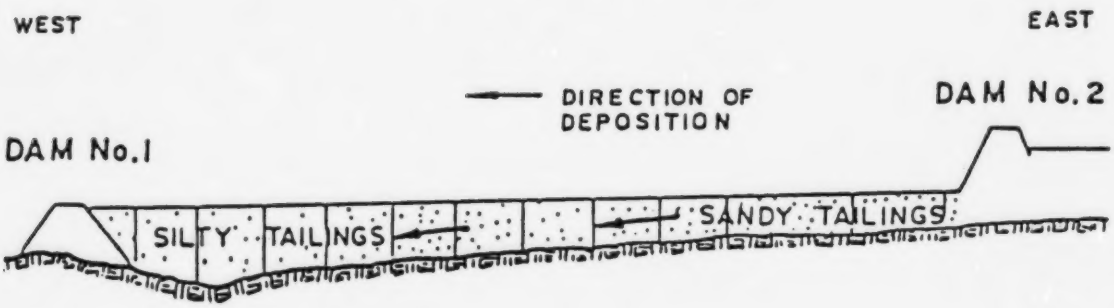
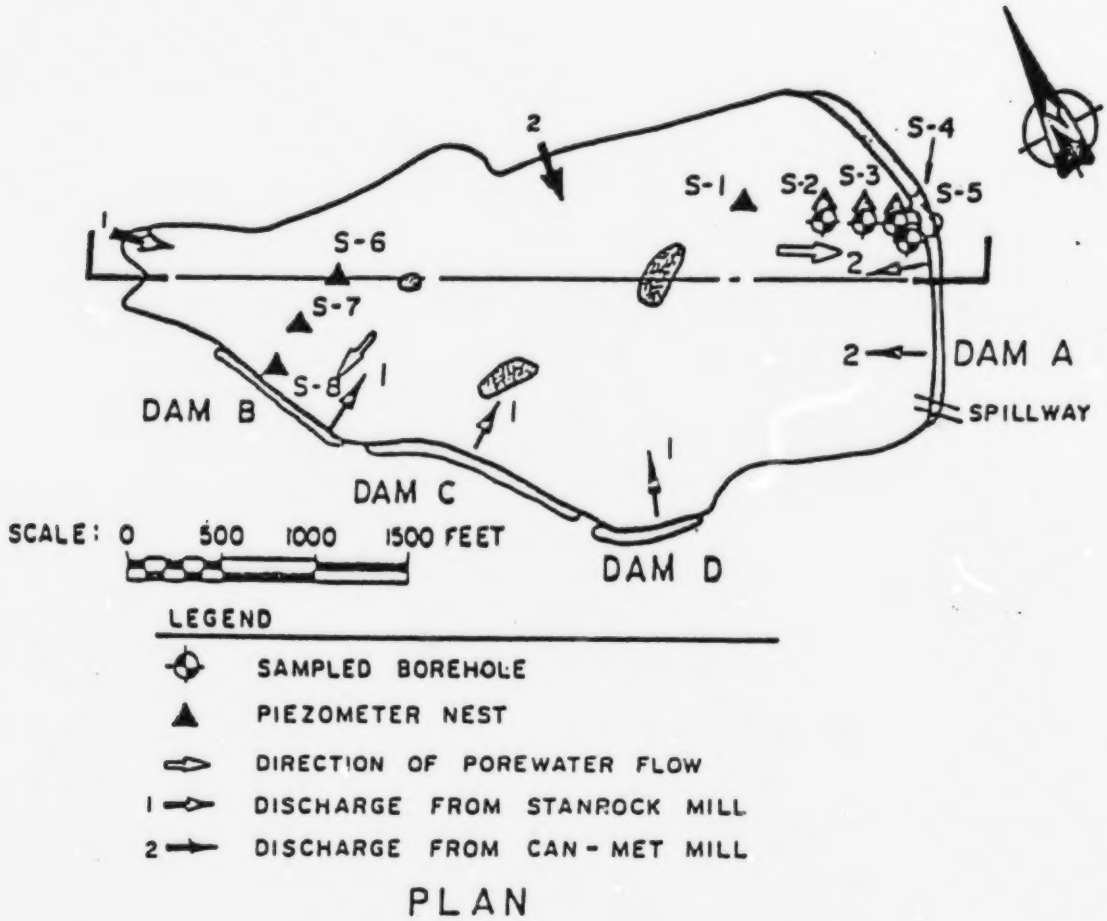


Figure 5. Williams Lake tailings site (2).



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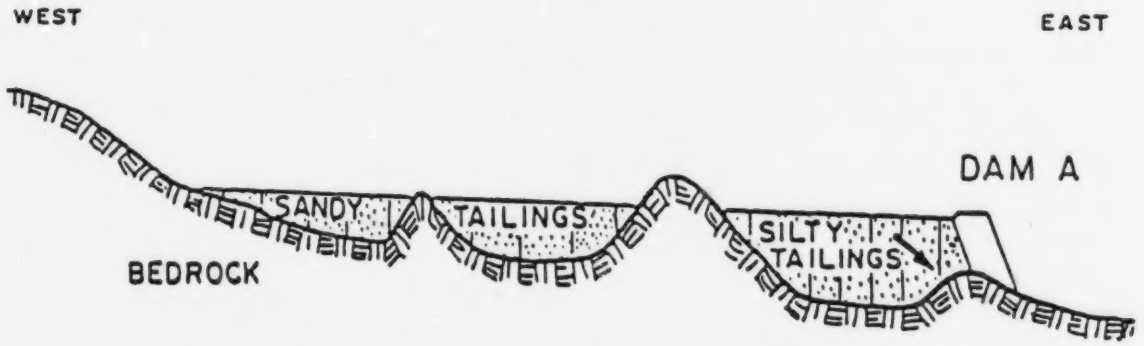

SECTION

Figure 6. Stanrock tailings site (2).

3.3 POROSITY

Porosity is well understood and easily measured. No research is necessary for the needs of this program.

Porosity

The percentage of open space to the whole space.

Total Porosity

The ratio of the volume of voids to the volume of solids generally determined by volumetric-gravimetric technique.

Effective Porosity

The amount of interconnected pore space available for fluid transmission, expressed as a percentage of the total volume.

Porosity of uniformly sized spheres varies from a maximum of 47 per cent for the least stable packing to a minimum of 26 per cent for the most stable. Natural materials are not spherical and, therefore, their measured porosities extend beyond these limits (10).

If the volumetric composition of a sand is of minerals that are relatively stable, no significant dissolution of the grains or changes in porosity and permeability will be of concern (10).

3.3.1 Porosity and Hydraulic Conductivity

The porosity can be an important controlling influence on hydraulic conductivity. In sampling programs carried out within deposits of well-sorted sand or in fractured rock formations samples with higher porosities generally also have higher hydraulic conductivities. Unfortunately, the relationship does not hold on a regional basis across the spectrum of possible rock and soil types. Clay-rich soils, for example, usually have higher porosities than sandy or gravelly soils, but lower hydraulic conductivities.

3.3.2 Porosity and Void Ratio

The porosity (n) is closely related to the void ratio (e), a parameter which is widely used in soil mechanics (4). The void ratio is defined as:

$$e = \frac{V_v}{V_s}$$

where V_v = volume of voids

V_s = volume of solids

And e is related to n by:

$$e = \frac{n}{1-n} \quad \text{or} \quad n = \frac{e}{1+e}$$

where e normally ranges from 0 to 3.

3.3.3 Measurement

Porosity can be measured in principle by saturating a sample, measuring its total volume, V_T, weighing it and then oven-drying it to constant weight at 105°C. Knowing the density of water, the weight of water removed is converted to a volume. This volume V_v is the volume of the void space:

$$n = \frac{V_v}{V_T}$$

where V_T = total volume

V_v = volume of voids

In practice it is quite difficult to completely saturate a sample of given volume. It is more usual to make use of the relationship (10):

$$n = 1 - \frac{P_b}{P_s}$$

where P_b = bulk density

P_s = particle mass density

The bulk density is the oven-dried mass of the sample divided by the volume. The particle density is the oven-dried mass divided by the volume of the solid particles, as determined by a water-displacement test. In cases where great accuracy is not required, $P_s = 2.65$ g/cm^3 can be assumed for most mineral soils.

A void ratio value of 0.9 is representative of the initial densities usually obtained by hydraulic placement of tailings sands (6). In the literature the value used for tailings porosity was 0.3 (expressed as a fraction).

To predict the flow of a fluid in porous media, in time and space, the porosity and permeability are primary parameters that must be specified.

The factors and their relative importance in determining the porosity of unconsolidated natural deposits are (10):

- Absolute grain size
- Non-uniformity in size of grains
- Proportions of various sizes of grains
- Shape of grains
- Method of deposition
- Compaction during and following deposition
- Solidification.

With the knowledge of the tailings porosity, water content, thickness, overburden depth and also the Ra-226 concentration in the tailings, the Rn-222 flux from the pile may be reasonably estimated (6).

3.4 DENSITY

Density measurements are simple to perform. In-situ measurements can be made reliably by the nuclear probe method providing that calibration tests are completed on the specific material in question.

The following definition of density and specific gravity are cited from the American Society of Testing and Materials Standards (ASTM No. 12-70); in all such determinations, the temperature of the material and water used in the test must be specified (10).

True Density

The mass of a unit volume of a solid material where the volume of only the impermeable portion is considered. This can be represented simply as:

$$T.D. = \frac{\text{mass}}{(\text{volume} - \text{voids})}$$

True Specific Gravity

The ratio of the mass of a unit volume of material to the mass of the same volume of gas-free distilled water. For solid material, the volume used is that of the impermeable part.

Apparent Density

The weight in air of the impervious part of the unit volume of rock.

Apparent Specific Gravity

The ratio of the weight in air of a unit volume of material to the weight in air (of equal density) of an equal volume of gas-free distilled water:

$$A.S.G. = \frac{\text{dry weight in air}}{(\text{dry weight in air} - \text{submerged weight})}$$

Bulk Density

The weight in air of a unit volume of a permeable material including both permeable and impermeable voids normal to the material. This can be expressed as:

$$B.D. = \frac{\text{weight}}{\text{volume including all voids}}$$

or

$$B.D. = \frac{\text{weight}}{\text{unit volume, including water}}$$

The bulk density frequently is considered to be synonymous with the apparent density (ASTM Standard B243-65 and E12-70).

Bulk Specific Gravity

The ratio of the weight in air of a unit volume of a permeable material (including all voids) to the weight in air (of equal density) of an equal volume of gas-free distilled water:

$$B.S.G. = \frac{\text{dry weight in air}}{\text{(saturated weight - submerged weight)}}$$

Grain Density

The mass of a unit volume of grains:

$$G.D. = \frac{\text{mass of grains}}{\text{volume of grains}}$$

It can be calculated by:

$$Pg = \sum_{i=1}^{i=n} P_i V_i$$

where: n = the number of minerals in the rock

P_i = the grain density of each mineral

V_i = the volume of each mineral

Determination of the grain density minimizes the effects of pore space and open fractures common to rocks (10).

Other more simple definitions found in the literature follow:

Bulk Density

The oven-dried mass divided by its field volume.

Particle Mass Density

The oven-dried mass divided by the volume of the solid particles, as determined by water-displacement test. In cases where great accuracy is not required P_s = 2.65 g/cm^3 can be assumed for most mineral soils.

Mass Density of a Fluid

Defined as the mass per unit volume.

Weight Density of a Fluid

Defined as the weight per unit volume.

Specific Gravity

The ratio of the density (or specific weight N/m^3) to that of water. For water G = 1.0, for most soils and for rock G = 2.65.

A confusing element in analysing published data on density is that some authors do not differentiate between "true", "apparent", and "bulk" determinations. Also, if the measurement units are not stated, it must be assumed that the data represents specific gravity. If a value is given as 2.55 gm/cm^3, the value is clearly that of density - true, apparent or bulk; however, if merely 2.55 is stated, then it most probably is specific gravity. In analysing thousands of data sets in the literature, the preceeding assumption was adopted (10).

The use of density as an index property or as an independent variable to predict rock properties is difficult because density values can be affected by temperature, pressure, and the amount and type of fluid saturating the media.

3.5 DISPERSION (Dispersivity)

As the contaminants move along the groundwater flow paths, concentrations decline because of mixing of the contaminated groundwater with the ambiant groundwater and because of geochemical processes. These mixing processes are referred to as dispersion. Dispersion in groundwater can occur by diffusion, hydrodynamic processes due to aquifer anisotropies, and chemical processes. Dispersivity is a parameter which characterizes the spreading or dispersion attributable to the medium being considered.

The effects of molecular diffusion, included in dispersion are usually neglected except at low advection rates. Dispersion also includes the mixing resulting from velocity differences affected by microscopic and macroscopic variations in porous media. Dispersion resulting from variations in velocity is referred to as mechanical or hydraulic dispersion. Hydrodynamic dispersion is the tendency for the solute to spread out from the path that it would be expected to follow according to the advective hydraulics of the flow system. It causes dilution of the solute. It occurs because of mechanical mixing during fluid advection and because of molecular diffusion due to the thermal-kinetic energy of the solute particles (5).

Dispersion occurs in all circumstances where contaminants migrate in groundwater, but it is generally not a cause of very large decline in contaminant concentration. For sub-surface travel distances of hundreds of meters in permeable deposits of sand or gravel, dispersion rarely causes contaminant concentrations to diminish by more than a factor of 10. When contaminated groundwater enters streams or lakes, additional diminution in concentration takes place as a result of mixing with waters from other sources. Although dispersion in the groundwater zone can be counted upon to cause a decline in contaminants at exit points, the decline is generally not large enough to act as a dominant influence on the attenuation of the contaminants in the sub-surface domain (11).

3.5.1 Dispersion and Directional Properties

Dispersion has directional properties. The spreading of solute in the direction of flow is referred to as longitudinal dispersion and spreading perpendicular to the direction of flow lines is referred to as transverse dispersion.

For porous media, the concepts of average linear velocity and longitudinal dispersion are closely related. Longitudinal dispersion is the process whereby some of the water and solute molecules travel more rapidly than the average linear velocity and some travel more slowly. The solute, therefore, spreads out in the direction of flow and declines in concentration.

Mechanical dispersion in the transverse direction is a weaker process than dispersion in the longitudinal direction, but at low velocities where molecular diffusion is the dominant dispersive mechanism, the coefficients of longitudinal and transverse dispersion are nearly equal.

Longitudinal dispersion is assumed to begin with the downgradient movement.

3.5.2 Dispersion Modelling

There are some simple models in the literature on longitudinal dispersal, dilution, and chemical changes accompanying dispersion (12). Anderson has presented an excellent review of simulation modelling of contaminants in groundwater flow systems involving dispersion (10).

It is stated that the coefficient of dispersion is of ultimate importance in the analysis of contaminant transport.

3.5.3 Dispersion and Groundwater Velocity

In many geographic regions excessive uncertainty in advection rates and dispersion coefficients can be avoided if the hydrogeological zones selected for sub-surface waste emplacement have very small groundwater velocities. In this case, mechanical dispersion is negligible relative to molecular diffusion and the mass flux of contaminants is primarily a diffusive flux with little contribution from advection. If dispersivity is large, contaminants can spread to occupy a portion of the flow system many times larger than would be the case in the presence of advection alone.

3.5.4 Values and Measurement

Dispersivity values obtained from column tests on disturbed or undisturbed samples of unconsolidated geological materials yield values in the range of 0.01 to 2 cm (5). Values of longitudinal dispersivity depend on grain size and grain-size distribution and are independent of grain shape, roughness and angularity.

Experiments using tracer and/or contaminants spreading in the field as a result of dispersion normally give greater values than indicated by laboratory measurements. This difference is normally attributed to the effects of heterogeneities on the macroscopic flow field. Since most heterogeneities in geological materials occur at a larger scale than can be included in borehole samples, dispersivity values from tests on small samples can be viewed as representing a property of the medium, but at a scale of insufficient size for general use in prediction of dispersion in the field. Studies of contaminant migration under field conditions require dispersivity measurements in the field. Although this premise is generally accepted, there is little agreement on the most appropriate methods for determining field dispersivity. Very few field dispersivity test results are reported in the literature (5).

3.6 DIFFUSIVITY

Despite the following definition of hydraulic diffusivity, oxygen diffusivity and diffusivity in general are not specifically documented in the literature.

The theoretical definition of hydraulic diffusivity is transmissivity over the storativity and is not widely used in practice. The concepts of transmissivity and strorativity were developed primarily for the analysis of well hydraulics in confined aquifers, but they lose their meaning in many other groundwater applications. If a groundwater problem has three-dimensional characteristics, it is best to revert to the use of hydraulic conductivity and specific storage; or perhaps even better, to the fundamental parameters permeability, porosity and compressibility (5).

Murray has defined the air entry value as the amount of suction pressure that must be imposed on the soil before air will enter the sample (13). The value can be roughly considered as the minimum depth of the water table. The tailings samples require a great deal more pressure to remove water and to obtain air entry than soils.

Another author referred to the oxygen diffusion effect on pyrite oxidation (2). The oxidation of pyrite in the tailings occurs in the unsaturated zone of the tailings close to the surface where oxygen from the atmosphere can readily flow or diffuse downward into the tailings. The low solubility of oxygen in water and the low diffusivity of oxygen through water will substantially reduce the availability of oxygen and, therefore, the oxidation of pyrite in water saturated or near-saturated tailings. Investigations in the tailings unsaturated zone by Smyth (14) determined that high oxygen concentrations in the pore gas are restricted to the upper 1.0 m. The low pH porewater in the unsaturated zone occurs close to the surface. These results suggest that pyrite oxidation predominantly occurs close to the surface of the tailings (2).

Some experts believe that oxygen diffusivity is not a limiting factor on pyrite oxidation because the oxidation will occur with only 0.01 to 0.02 per cent of oxygen in the tailings (15).

3.7 SOIL TEMPERATURE

Seasonal variations in soil temperatures have been studied by Grenier and Bigu and some models have been produced including one by Ouellet that is easy to use (16-18). However, this model poorly predicts conditions in the top 10 cm and cannot be used for soil temperature variations in a northern climate.

No studies appear to have been done in the cold climates of northern Saskatchewan and studies made on other areas cannot be used to predict effects in this location. Tailings are a very different composition to the types of agricultural soil that have been most widely studied to date.

Some studies have shown that temperature has only a small effect on the solubility of radium and non-radioactive substances (19, 20).

From this work it appears that there is very little seasonal changes. With the temperatures experienced during the summer (5 to 45°C) there was little change in the extent of leaching with temperature.

There were no studies found that considered the effects of temperature below the freezing point. Such information would be useful for the Canadian situation as temperatures are below 0°C for most of the winter in areas where uranium tailings are present.

Furthermore, geochemical stabilization of the surface of a pyritic tailings pile should theoretically isolate the underlying tailings from atmospheric and higher surface temperatures and, thereby, reduce the pyrite oxidation rate (2).

The effect of changes in temperature will become a greater concern in the long-term. This will be discussed in the following section.

4. DISCUSSION AND CONCLUSION

The factors influencing the mobility of the contaminants and the low pH fronts in pyritic tailings are primarily chemical (i.e. chemical reactions). The most important aspects of the groundwater pathway are geochemical and hydrochemical and not geochemical and hydrological. The literature review showed that retardation and flow rates are dependent upon adsorption, ion exchange, precipitation, co-precipitation and possibly other chemical reactions. Physical parameter influence is mainly on the flow rate of the tailings porewater and groundwater. If physical parameters and atmospheric conditions (mainly temperature and pressure) are favourable for high groundwater velocity the removal of contaminants will be faster in the long-term. However, if groundwater flow is slow, the amount of contaminants going into solution will be lower and the process of contaminant retardation in the tailings and in the groundwater regime outside the tailings area will, in a specific pore volume, last longer.

It is this aspect of "time" that will be influenced by the physical characteristics of the tailings and of the surrounding soils. The processes that will effect flow rates of contaminants and consequently the relative time it will take the contaminants and the pH fronts to be displaced is mainly due to the retardation effects of calcite and other chemical reactions (8). The consumption of calcite causes a rise in the pH and, thus deposition of contaminants on the solids of the porous media. The contaminant plumes, because of retardation processes, advance only at a fraction of the porewater or groundwater flow rates. If the groundwater flow rate is high, the calcite will be consumed more rapidly and the low pH plume will advance. It is said that less than one weight per cent calcite is still sufficient to neutralize pH for 200 years. The depletion time calculation for calcite is dependent upon pH, porosity, specific gravity, and groundwater flow rate.

The measurement, and the comprehension of hydraulic-conductivity, hydraulic gradient, porosity and density does not seem to be a problem. The information on basic understanding is available in the literature and is considered sufficient (except for data). However, throughout the review, it was mentioned that diffusivity and dispersivity are not sufficiently understood and are difficult to measure.

Diffusivity is difficult to measure because of factors such as chemical and bacterial reactions that influence oxygen diffusivity. For example, it is not just porosity that will effect oxygen diffusivity in the air in the unsaturated zone and in the porewater.

Dispersivity is an important parameter in mathematical modelling. However, it is a difficult parameter to use because it is impossible to assign a fixed dispersivity value for a specific area. Dispersivity changes with distance from the source and also with time. It has not been possible to measure consistant values from field and laboratory work.

Laboratory measurements are usually lower than those taken in the field. The postulated reason for this difference is the mass difference between a laboratory sample and the in-situ measurement (8). This problem has only been addressed recently (1982-1984). Some dispersion modelling has been attempted, but with little success so far. Nonetheless, work is continuing at the University of Waterloo (8).

Soil temperature has been studied by Grenier and Bigu for the moderate climate of the Elliot Lake area (16). The recommended model does not predict accurately temperatures in the top 10 cm, but this is considered insignificant. Seasonal variations of soil temperature in moderate climates and their effects on chemical reactions in the tailings and in the groundwater systems are not important in the short- or medium-term (8).

The annual mean soil temperature is said to be sufficient for modelling use. However, in the long-term, if groundwater flow is slow because of physical characteristics of the soil aquifer and, thus if the movement of contaminants is slowed, the temperature becomes an important factor. If, over the long-term, the average yearly mean temperature were to rise because of transformation in climate of the earth, the migration of contaminants and low pH could be enhanced.

Little work has been done on the severe weather conditions that exist in northern Canada. It is quite likely that under these severe conditions, temperatures within the tailings will have a significant effect on contaminant movement. It is possible that by manipulating the tailings site conditions, the tailings can be retained in a frozen state on a permanent or semi-permanent basis, thus retarding the impact of the tailings site on the environment.

It could be important to consider long-term temperature effects on soil temperatures and on chemical reactions in tailings and groundwater regimes. It would be of somewhat lesser importance, however, if the contaminants do reach surface water systems and/or the general population in relatively short-time periods. In this case the temperature elevations would probably only increase the contaminant outflow concentration. The concern would be that the maximum concentration of contaminants in solution would become greater than the maximum allowable concentration. It is known that the earth has had many climatic changes in its lifetime and it would thus be important to consider these changes as much as reasonably possible in the future.

An understanding of the interdependencies of the physical parameters was the major objective of this review. The subject was investigated and certain relationships were discussed. It was concluded that the physical parameters would be considered independent for modelling. However, this conclusion was not unanimously agreed upon and the answer still remains somewhat unclear (8, 15, 21, 22).

During the literature review, sections on natural events such as earthquakes, floods and wind storms were observed. These subjects were not discussed in this report, but references are tabled for completeness (23, 24).

5. EROSION

The erosion of tailings is considered a potential major problem. Erosion models are said to be scarce and poor and erosional processes to be somewhat unpredictable (25).

M. A. J. Williams (26) has defined the five major processes of erosion as:

a) Mass movement, which includes soil creep (slow compared with slope wash or overland flow erosion), surface rock creep which increase in a linear manner with increased slope angle and landslides (not common, although rock falls do occur on the escarpments).

b) Rain splash erosion, a major mechanism for soil removal by high intensity rainfall, particularly at the start of the wet season, when the rate may be 20 to 40 times higher than later in the wet season.

c) Overland flow, essentially a function of slope length, soil type, precipitation intensity and duration, and surface cover. This is much higher in footslope areas than on steep, rocky hillslopes. It is up to five times greater than creep.

d) Throughflow, more active in the wet season, when profiles have been deeply infiltrated by water and when downslope movements of water do appear to increase rates of footslope erosion.

e) Gully erosion, channel formation by runoff concentration in tracks made by animals and vehicles, and representing a much accelerated form of denudation.

The causes of gully formation on the cover of a tailings site or on embankments would be a combination of steep slopes, the concentration of runoff, and the erodibility of the soil. Because gullying is more severe in areas of steep slopes, it is unlikely that significant gullying of the cover would occur independently of serious gullying or instability of the retaining embankments. If gullies form directly on the cover without having been started in the embankment the effects would probably be local (23).

Any increase in precipitation, decrease in infiltration capacity of the soil, or loss of vegetation on the cover could initiate gully formation. On the other hand, cementation of the cover material by natural weathering could retard the formation of gullies.

Differential settlement may cause cracking or depressions that would concentrate runoff at specific points on the embankment increasing the potential for gullying. Gully formation on the cover could concentrate runoff and cause gully formation on the embankment. The reverse is also true. Any change in climate or vegetation cover could increase runoff or decrease stability and lead to gully formation on the embankment. If the toe of the embankment is eroded and steepening of the slope results, risk of gully formation will increase (23).

The probability of gully formation on the cover of an impoundment other than due to headward progression of embankment gullies will generally be low. The chance of gullies progressing from the embankment will be higher than for the cover because of steeper slopes.

Over long periods of time the likelihood for gully formation on the cover and on the embankment will be relatively high. If environmental and geomorphological conditions do not change, and if the cover has been well constructed, it would be safe to assume that gullying will not be a problem. However, over the medium- to long-term period, and even within short intervals, it is unrealistic to expect conditions not to change.

The climate is likely to change over the long-term. Drought may occur for some years followed by years of abnormally high precipitation. A change in precipitation and temperature will affect sediment yield and rainsplash erosion (23, 27, 28). Major climatic changes may not occur within the short-term period, but could certainly be expected within a medium-term period.

Dr. T. A. Shepherd clearly states that there is no mathematical model that predicts gully formation. Schumm (27) and Brice (29) have suggested that by study of natural gully formation in a specific area some indication of a stable versus unstable condition can be established. In addition, evaluation of calculated soil losses using the Universal Soil Loss Equation for different conditions could provide some insight into the potential for gullying.

Some modelling work on gully formation has been completed although it would seem to have met with little success to date. Steeve Abt and Stanley Schumm of Colorado State University, Fort Collins, Colorado, David L. Rees of Nipissing University College, North Bay, Ontario were identified as possible future references for modelling work (30).

6. ADDITIONAL NOTE

During the course of this work, a specific pathway of contaminant transport to humans was found to be missing. The pathway in question would take radon emanation and dust particles to animals by inhalation and then to man by ingestion, and/or dust particles to soils and plants to animals by inhalation then to man by ingestion. This pathway is mentioned for the record.

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APPENDIX A

TAILINGS SITE DESCRIPTIONS

Elliot Lake - General

The tailings sites that have been developed since the start of mining of the Elliot Lake ore are shown on Figure A-1.

Figure A-1. Elliot Lake area tailings sites (A-1).

Nordic (A-2)

The Nordic mine, located approximately 6 km south of Elliot Lake, Ontario, was operated by Rio Algom Limited from 1957 to 1968. Approximately 11,000,000 tonnes of neutralized tailings were discharged to the tailings area which is contained by rockfill dams on three sides and a hill on the north side. The mill circuit consisted of acid leaching, ion exchange and ammonia precipitation which is the typical uranium circuit operated in the Elliot Lake area.

The complete tailings site was revegetated by Rio Algom Limited during the period 1971 to 1978. The surface characteristics of the Nordic main and the Nordic westarm tailings sites are shown in Figures A-2 and A-3.

Due to the accessibility of this tailings area numerous investigations have been carried out at this site.

Some of the major comments for this area include:

- the tailings are pyritic (5-6% FeS_2) and acid generating;
- primary concern is the acidic seepage with its inherent heavy metals content and radionuclides;
- the zone front of low pH seepage water moves at a slower rate than the groundwater front due to neutralization effects of calcite in the sand aquifer;
- in front of the low pH zone the concentrations of heavy metals and radionuclides are acceptable;
- behind this front the pH is low and high concentrations of heavy metals and radionuclides exist;
- precipitation and adsorption reactions in the pH control zone plays an important part in the control of contaminants in the groundwater.

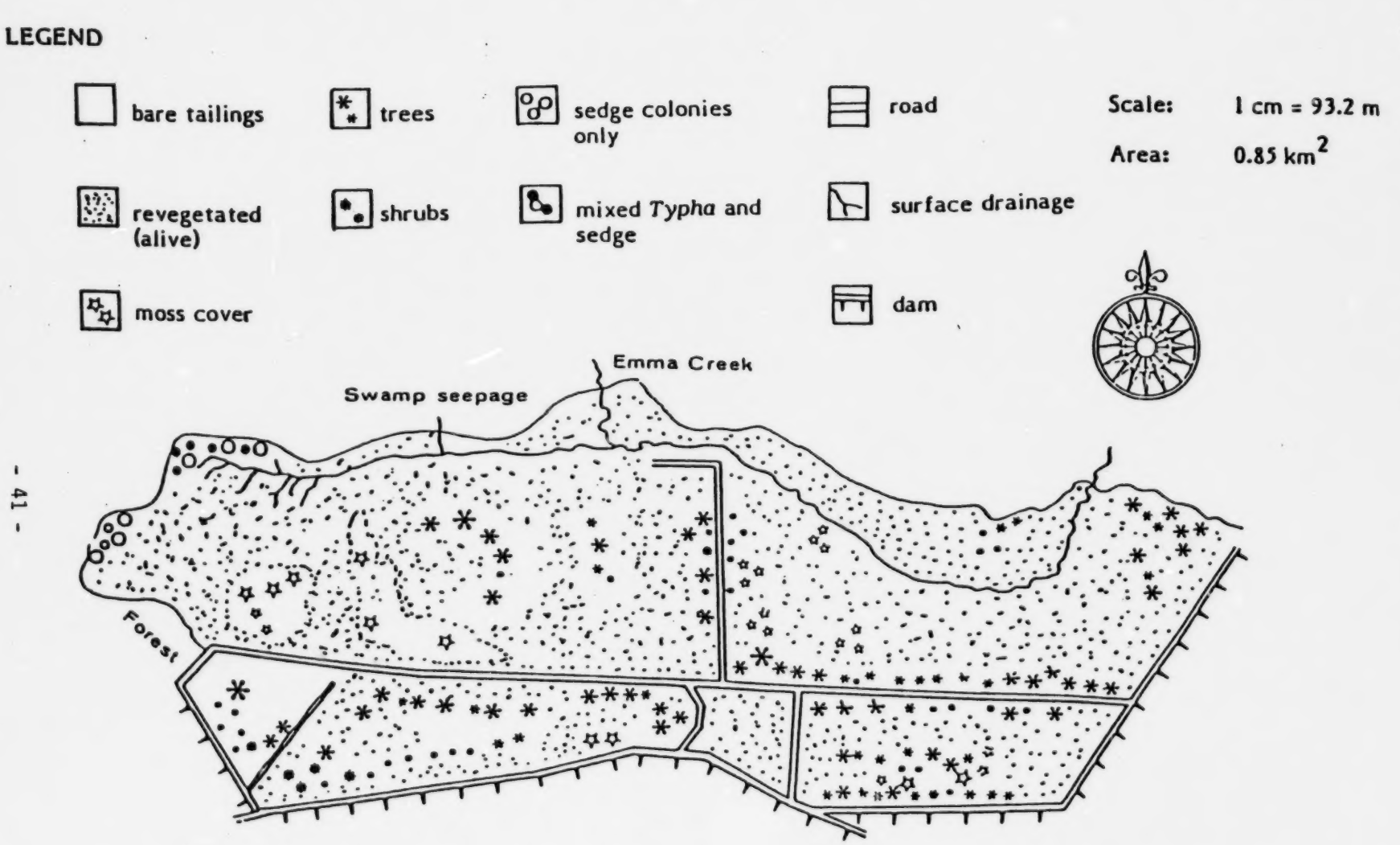

Figure A-2. Nordic main tailings site (A-3).

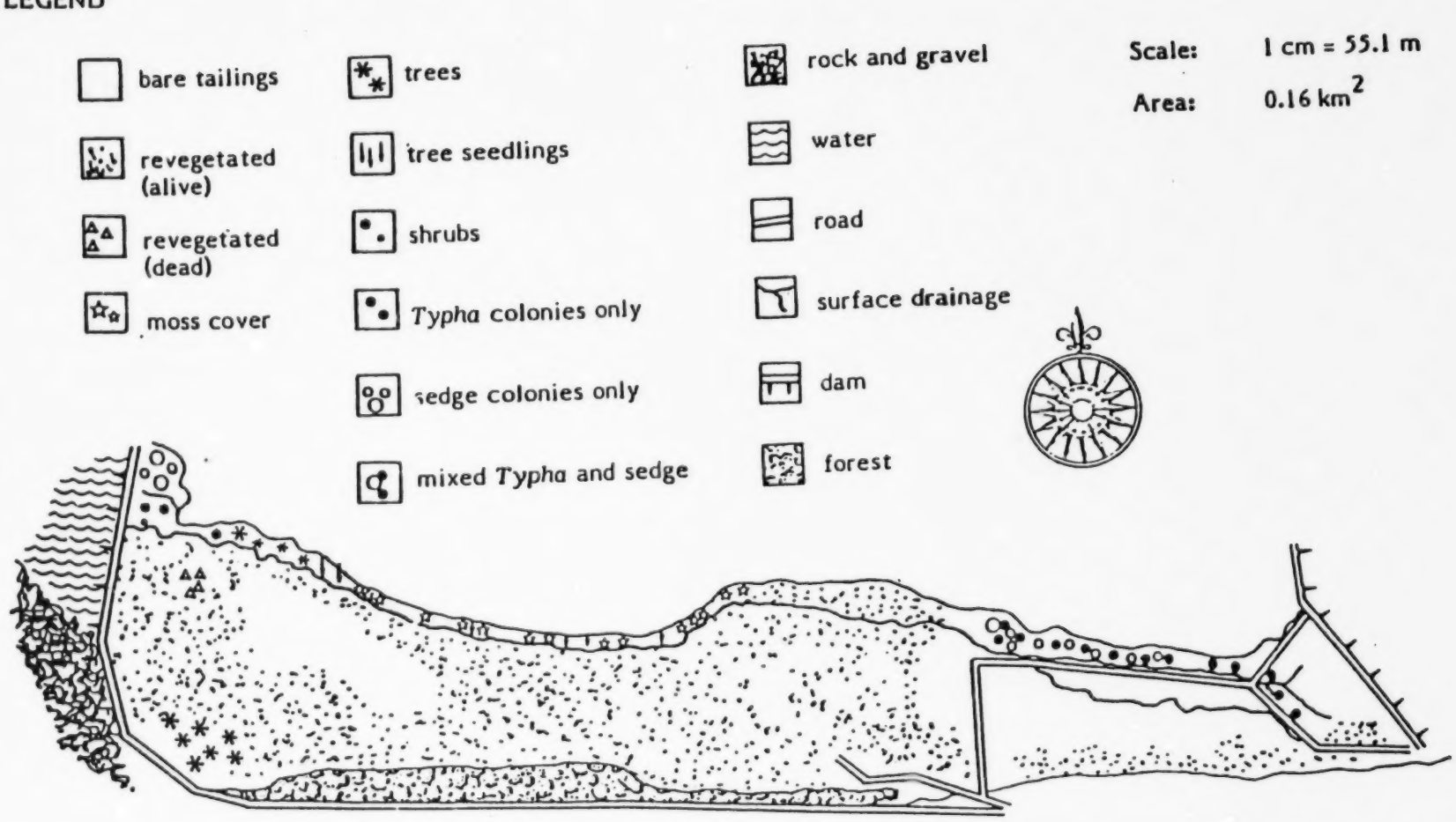

Figure A-3. Nordic west arm tailings site (A-3).

Stanrock (A-2)

The Stanrock operation was operated by Stanrock Uranium Mines Limited from 1958 to 1964 with approximately 5,800,000 tonnes of neutralized tailings being discharged to the tailings site.

An additional 2,200,000 tonnes of tailings were discharged to the Stanrock tailings site from the adjacent Can-Met mill operated by Can-Met Exploration Limited from 1957 to 1960.

The Stanrock operation continued to recover uranium with a bacterial leach operation from 1964 to 1969.

The tailings site is located north of Elliot Lake, approximately 12 km east of Highway 108. A dam failure on the east side of the tailings area has caused tailings to spread into an adjacent swampy area and creek. This dam is constructed of classified tailings sands.

The Stanrock main area was sprayed with lime in 1978 as a method of dust control. This has accelerated the encroachment of vegetation onto the tailings area.

The surface characteristics of the Stanrock main, the Stanrock Dam A and Stanrock Dam G area have been detailed by Kalin, 1982 (Figures A-4 to A-6).

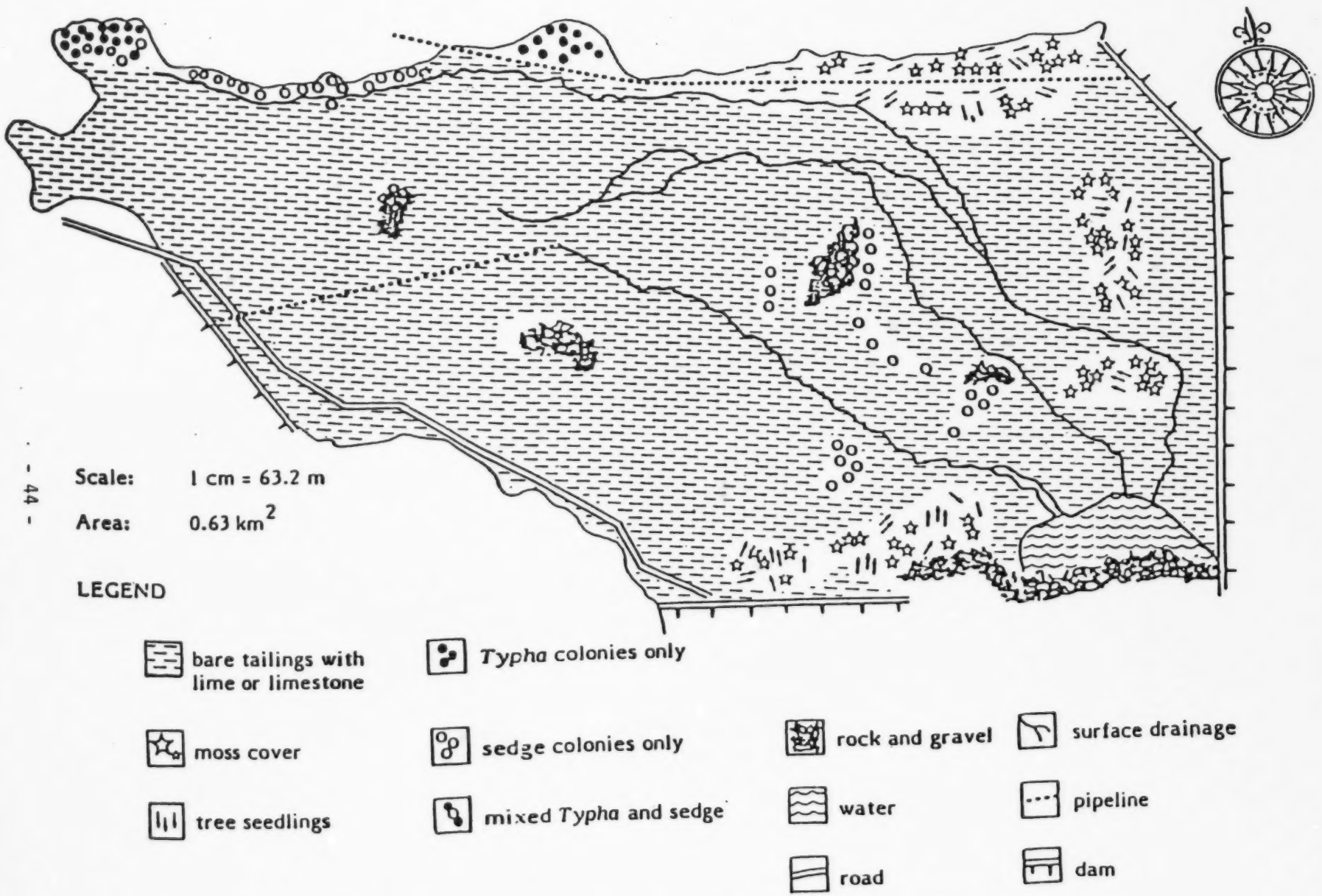

Figure A-4. Stanrock main tailings site (A-3)

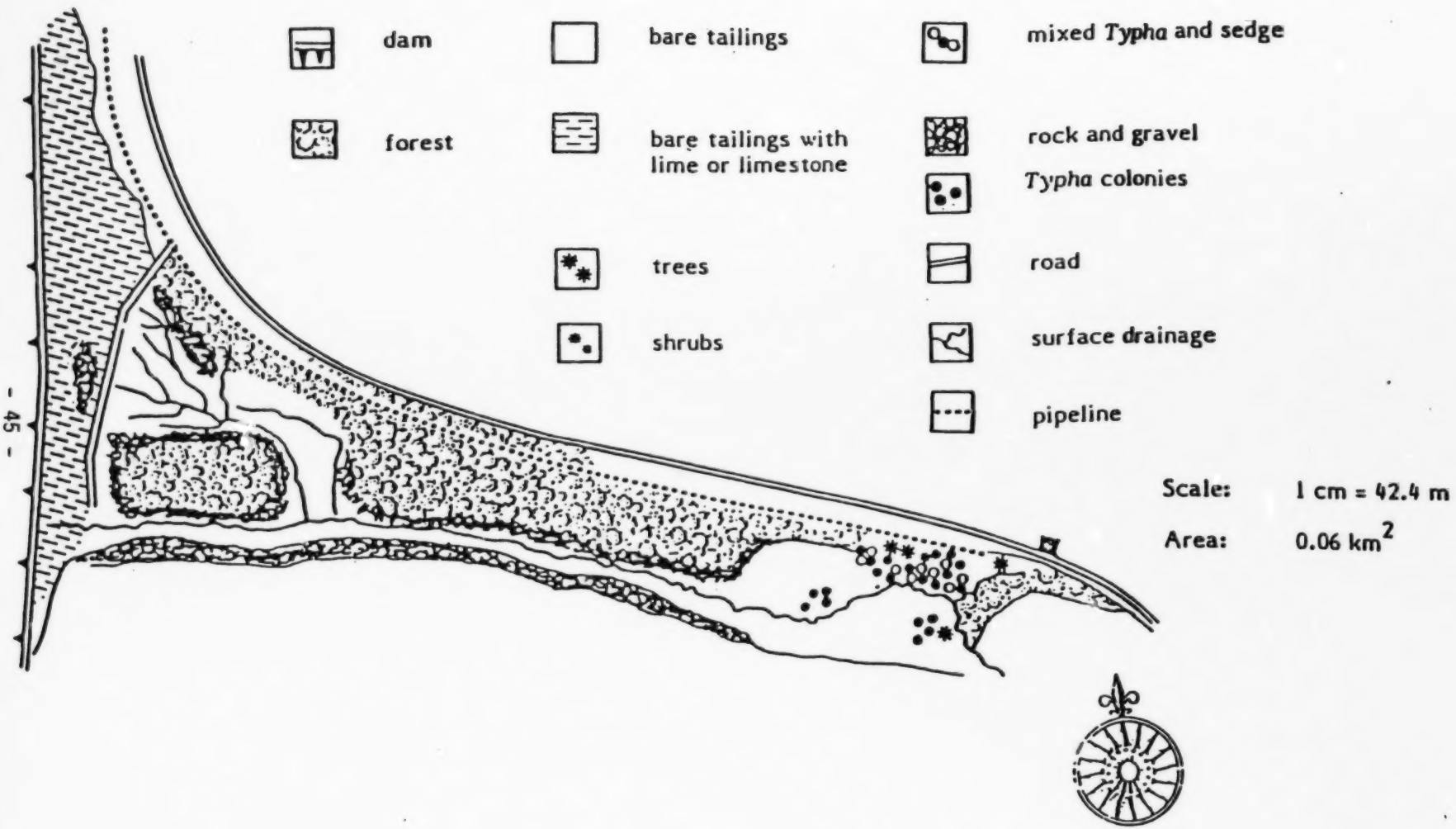

Figure A-5. Stanrock - Dam A tailings site (A-3).

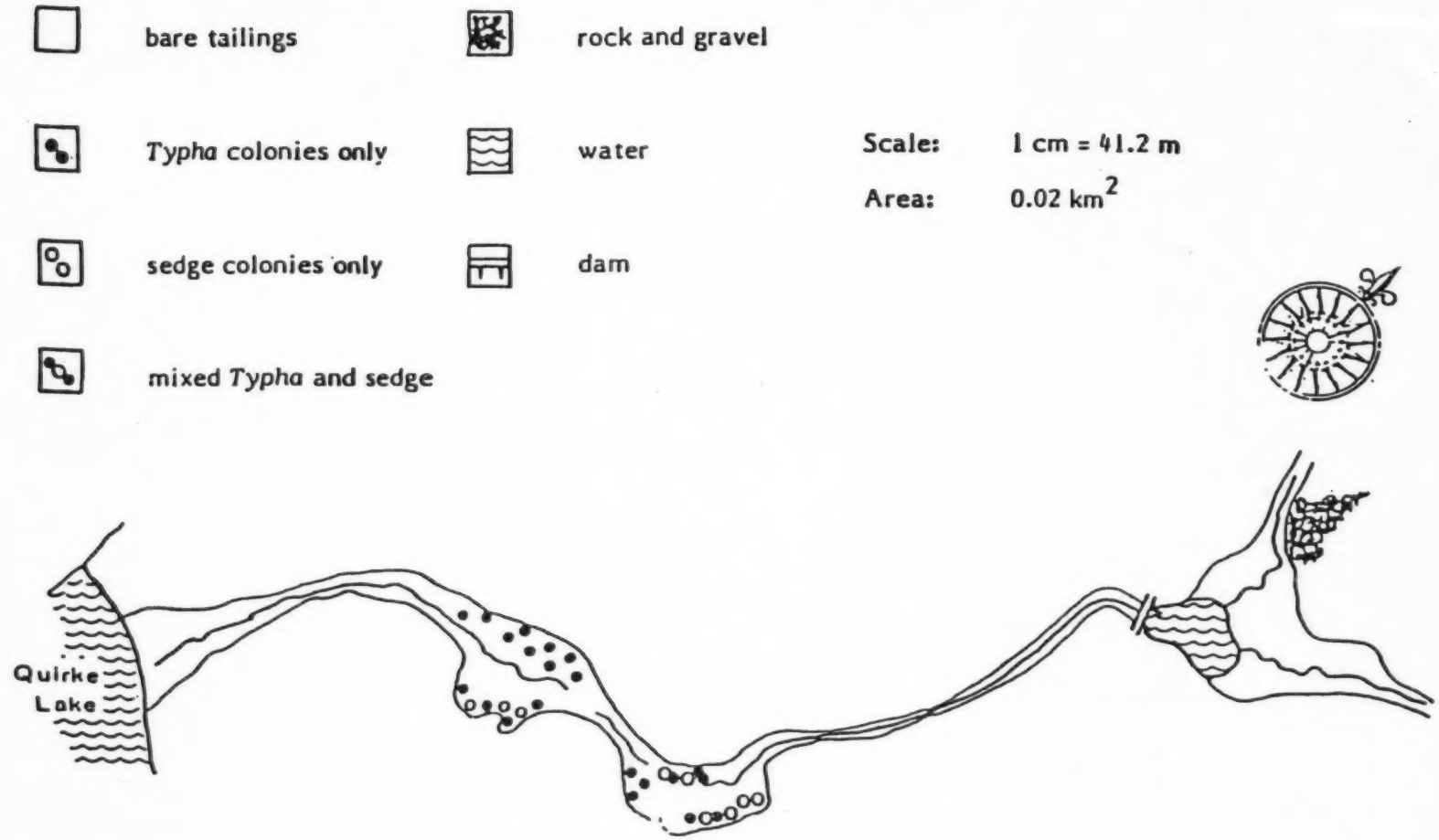

Figure A-6. Stanrock - Dam G tailings site (A-3).

Denison (A-2)

The Denison mine, operated by Denison Mines Limited, has been in continuous operation since 1957. The mill throughout has been increased in stages to the present capacity of 11,000 t/day. The mill circuit consisted of acid leaching, ion exchange and ammonia precipitation.

A total of approximately 40,000,000 tonnes of tailings has been deposited in the Williams Lake and Long Lake tailings basin (Figure A-7). The Williams Lake area is no longer used and some reclamation work has been carried out in the lower Williams Lake tailings site using a glacial till cover and vegetation with grasses and legumes. The upper Williams Lake area contains exposed tailings. The surface characteristics of the lower Williams Lake tailings site are described in Figure A-8.

The Long Lake basin is made up of the main dam at the west end and a dam at the east end. Tailings slurry is discharged at the east end which, in practice, results in the segregation of coarse and slimes with the slimes flowing to the west end of the tailings site.

Activities of interest at Denison include investigations of thickened tailings for increasing the capacity of the Long Lake basin and for providing a tailings surface compatible with revegetation, as well as the practice of mine backfill as a current mining method.

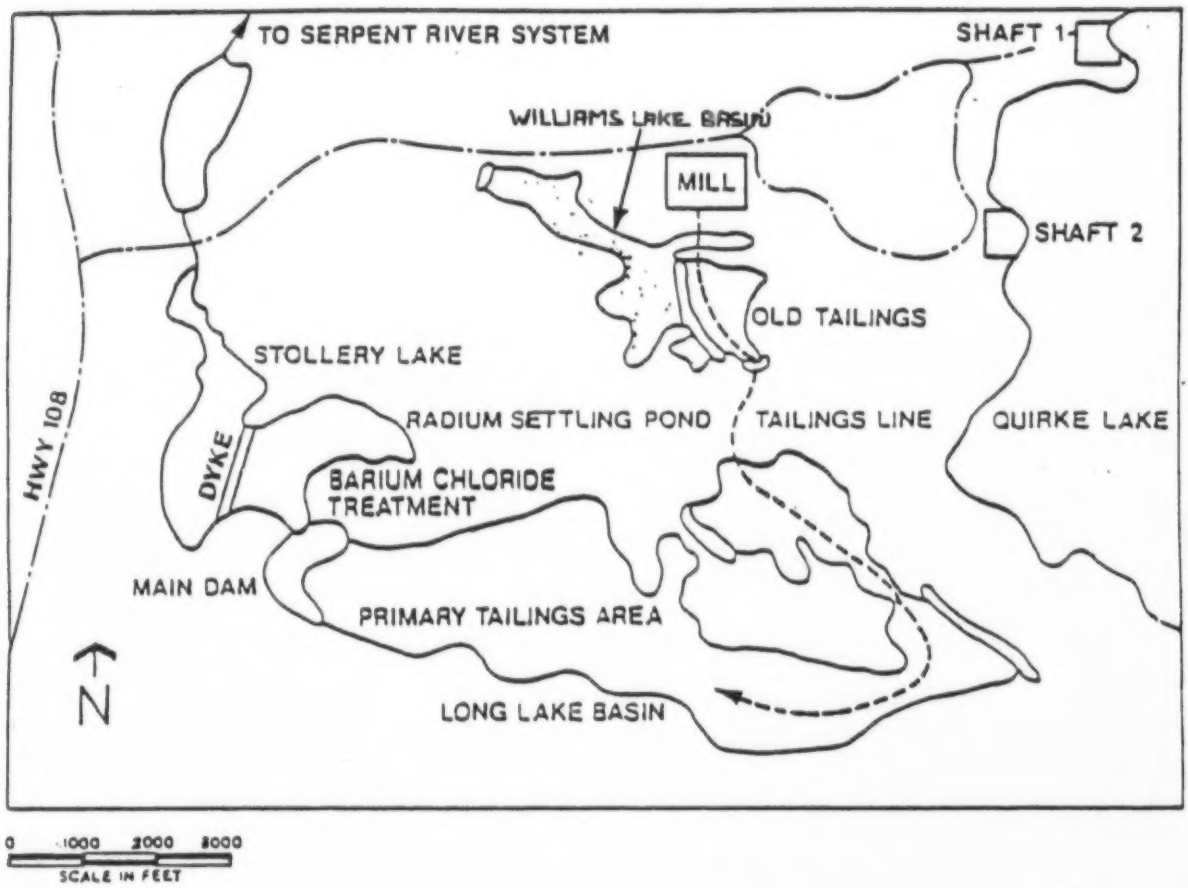

Figure A-7. Uranium tailings management system Elliot Lake, Ontario: Denison Mines (A-4).

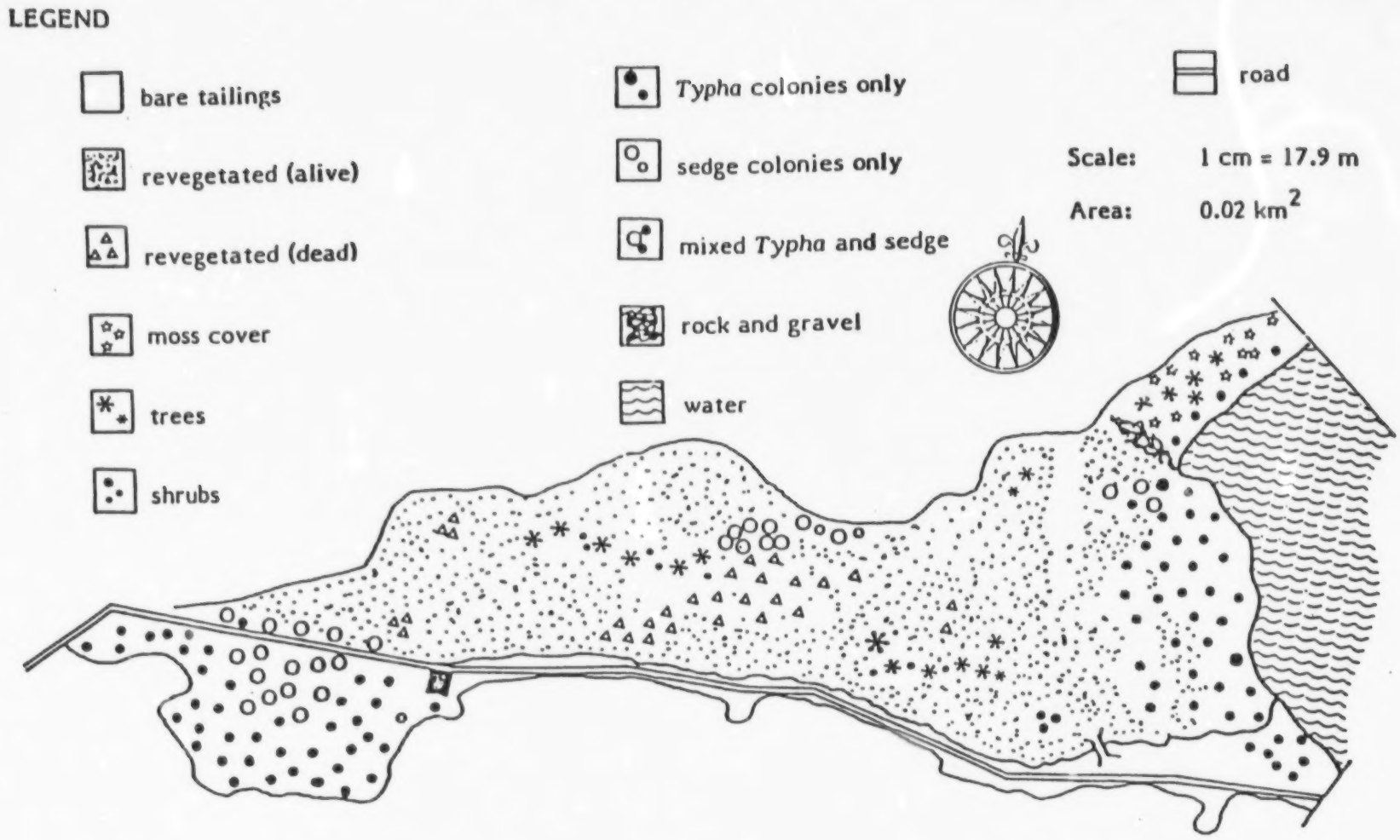

Figure A-8. Lower Williams Lake tailings site (A-2).

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